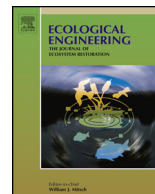




Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Floating treatment wetland aided remediation of nitrogen and phosphorus from simulated stormwater runoff

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ARTICLE INFO

Article history:

Received 15 February 2013

Received in revised form 24 July 2013

Accepted 20 September 2013

Keywords:

Urban stormwater

Agricultural runoff

Dissolved oxygen

pH adjustment

Temperature moderation

Aquatic macrophyte

Beemats

*Juncus effusus**Canna flaccida*

ABSTRACT

Floating treatment wetlands are potential alternatives to traditional constructed wetlands for remediating nutrient-rich water. This study examined the remediation efficacy of floating treatment wetlands planted with *Canna flaccida* and *Juncus effusus* in a replicated trough system over two growing seasons at two nutrient loading rates. Plant growth parameters were measured on a biweekly basis, and water quality parameters were monitored weekly. Plant shoots and roots were harvested at the end of the first growing season, and biomass was dried, ground, and analyzed for nutrient content. *Juncus* plants fixed 28.5 ± 3.4 g N per m² and 1.69 ± 0.2 g P per m², while *Canna* fixed 16.8 ± 2.8 g N per m² and 1.05 ± 0.2 g P per m². More N and P were fixed in the below-mat biomass of both species than in the above-mat biomass, thus whole-plant harvest may be a critical management strategy for floating treatment wetlands. During the first season, when nutrient addition rates simulated stormwater loading conditions, effluent nutrient concentrations were very low and averaged 0.14 ± 0.04 mg L⁻¹ total N and 0.02 ± 0.01 mg L⁻¹ total P. During the second season, nutrient-loading rate into treatment wetlands was doubled to simulate a more nutrient-rich runoff, and effluent nutrient concentrations averaged 0.79 ± 0.3 mg L⁻¹ total N and 0.12 ± 0.03 mg L⁻¹ total P. Floating treatment wetlands may prove most effective in low nutrient environments where it is necessary to polish water quality to extremely low P concentrations.

Published by Elsevier B.V.

1. Introduction

Stormwater runoff from urban or agricultural sources contains nutrient, metal, and chemical contaminants. If introduced into surface waters, these contaminants can negatively impact water quality, degrading ecosystem health. The ever-increasing scope of urban development along with the rise of the modern environmental movement has heightened public concerns about water quality and availability. These concerns have encouraged the implementation of regulations mandating water quality criteria and limiting nutrient releases into the environment (USEPA, 2010a,b). In an effort to facilitate adherence to present and future regulations and to prevent environmental damage, best management practices (BMPs) designed to reduce the negative impacts of stormwater runoff and runoff contaminants (nutrients and metals) have been developed and applied (Scholes et al., 2008).

Some commonly applied BMPs for runoff management include detention basins, retention ponds, wetland basins and channels, biofilters, and media filters (Leisenring et al., 2010). These technologies are effective at slowing runoff and have effectively reduced nitrogen (N), sediment, copper, and zinc levels in runoff water (Ellis et al., 1994; Leisenring et al., 2010; Scholes et al., 2008; Taylor et al., 2006; White et al., 2011). Despite the success of BMPs for other contaminants, these methods cannot achieve consistent phosphorus (P) removal (Dunne et al., 2012; Hoffmann et al., 2012; Pant et al., 2001). Thus, additional BMP technologies need to be developed to attain desired P removal rates and to reduce potential for environmental damage.

Floating treatment wetlands (FTWs) may be the most readily applicable BMP for further reducing phosphorous levels (Chang et al., 2012). Floating treatment wetlands have been successfully used to remove nutrients, metals, and glycol from stormwater runoff and wastewater (Chang et al., 2012; Chong et al., 1999; Headley and Tanner, 2006; Hubbard, 2010; Hubbard et al., 2004; Mohan et al., 2010; Nahlik and Mitsch, 2006; Tanner and Headley, 2011; Zhou and Wang, 2010). Unlike conventional free water surface and subsurface flow wetlands that are often used to remediate nutrient-rich waters, FTWs can be established within existing water retention infrastructure. As a result, FTWs do not require

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that additional land area be devoted to water treatment activities; thus FTWs are likely to have lower initial investment costs because extensive site work is not necessary for FTW installation (Winston et al., 2013).

Though similar to traditional constructed wetlands in many ways, FTWs rely on artificial buoyant scaffolds to support plant material. These floating scaffolds elevate plant crowns above the water level, permitting establishment of marginal, semi-aquatic, and aquatic species in deeper waters. Because the root systems of the species in the FTW are suspended in the water column rather than rooted into sediment or gravel substrate, the amount of root surface area in the water column is greater. This increased root surface area in the water column provides additional habitat for bacterial colonization (Stewart et al., 2008), potentially facilitating increased contaminant uptake and transformation in the water column. The suspended root masses of FTWs filter sediments from the water column while facilitating nutrient and metal removal (Tanner and Headley, 2011). This is a key difference between FTWs and traditional wetland systems where the bulk of contaminant processing occurs in the sediment or gravel matrix rather than in the water column (Edwards et al., 2006; Tanner and Headley, 2011).

Some FTW systems rely on active plant harvest (e.g. Beemats, a commercially available FTW scaffold) to facilitate additional nutrient removal and to limit internal nutrient cycling, while others do not utilize plant harvest (e.g. Biohaven™, a commercially available FTW scaffold) to assist with nutrient removal. Plant harvest facilitates removal of nutrients, especially P, from internal wetland cycling processes (Hoffmann et al., 2012; White et al., 2010). However, when plants are harvested, organic carbon is also removed, potentially limiting the amount of organic carbon available to support the growth of microbial communities which process N (Lin et al., 2002). Thus, there are benefits and potential downsides for management of FTWs in either an active or passive manner.

This paper summarizes two seasons of replicated experiments designed to characterize the N and P removal capacity of FTWs. These data provide baseline information needed to develop criteria for the use of FTWs as BMPs for nutrient remediation of urban stormwater or agricultural runoff. It is critical not only to understand the capacity of FTWs to remove nutrients, but also to understand potential ecological effects if they are deployed in settings where stormwater would reach organisms sensitive to changes in pH, dissolved oxygen, or temperature. The specific objectives of the present study were to:

1. quantify FTW-mediated nutrient removal from simulated runoff water at two nutrient loading rates,
2. quantify plant uptake of nutrients into both above- and below-mat biomass and characterize plant growth in different nutrient loading contexts, and
3. characterize the impact of FTWs on effluent physico-chemical parameters: dissolved oxygen, pH, and temperature.

2. Materials and methods

2.1. Experimental floating treatment wetland construction

Experiments were conducted during the spring-fall seasons of 2008 and 2009. The experimental units consisted of 3 experimental FTWs that were constructed in three troughs, two troughs with a surface area of 1.15 m² and a volume of 0.59 m³ and one trough with a surface area of 3.03 m² and a volume of 1.89 m³ (Fig. 1A). Troughs were initially filled with water from Lake Hartwell (Clemson, SC). Floating mats were Beemats. The Beemats FTW scaffold used for these studies were 1 cm thick, 60 cm × 60 cm buoyant

interlocking foam mat squares joined using 10 cm nylon connectors and secured with 3 cm locking washers to maintain raft integrity. Each mat section had ten (7.5 cm) pre-cut holes, which were spaced 12 cm on center. Each mat was designed to allow insertion of a plant contained in a specially designed aerator pot (Fig. 1B–D). *Juncus effusus* (Soft rush) and *Canna flaccida* (Golden canna) plants were placed in aerator pots and seated in the Beemats floating mats. The plants were 6.35 cm-diameter, rooted liners, which were established in a soilless potting substrate and supplied by Beeman's Nursery (New Smyrna Beach, FL). Experimental FTWs were installed in the flow-through troughs on April 14, 2008 and April 23, 2009. There were two plantings (2008 and 2009) for all troughs, as plants were harvested at the end of the 2008 study. Rafts were sized such that they covered 95% of the water surface (Fig. 1D).

2.2. Simulation of nutrient containing runoff

The experimental FTWs were treated with a continual flow of pond water spiked with nutrients beginning on May 2, 2008 and April 23, 2009. The simulated stormwater runoff solution was prepared by dissolving water-soluble fertilizer (0.10 g/L in 2008 and 0.20 g/L in 2009 of a 20N–2P–20K Nitrate Special Soluble Fertilizer, Southern Agricultural Insecticides, Inc., Hendersonville, NC) in water contained in a large, 2000 L round stock tank. The water soluble fertilizer was completely dissolved in water prior to addition to the stock tank to ensure uniform distribution throughout the stock tank. Flow of the simulated runoff solution from the stock tank into the 3.03-m²-control/mixing tank was regulated to supply continuous and consistent nutrient loading rates into experimental units. The simulated runoff solution was mixed continuously with additional lake water in the control/mixing tank and then flowed through the 2.5 m control tank before being collected for calibrated distribution within the treatment tanks. Simulated runoff solution flowed at 140 mL/min into the 1.15 m² treatment tanks and at 450 mL/min into the 3.03 m² tank, achieving a 3-day hydraulic retention time (HRT) for each experimental FTW unit. Rainfall was not measured, but the system was open to air and any rainfall impacts on nutrient presence were accounted for in water samples collected.

2.3. Water sampling and chemical analysis

For each experimental FTW ($n=3$), water samples were collected every 7 days, beginning 3 days after initiation of fertilizer addition. Plant size measurements (shoot height and root length) were collected every 14 days and were made on the same three plants per species per treatment unit over the sampling periods. Water samples (100 mL) were collected and analyzed for NH₃⁺-N (Orion Ammonia Electrode 95-12, Thermo Electron Corp., Beverly, MA), anions including NO₃-N, NO₂-N, and PO₄-P (Dionex AS10 ion chromatograph with AS50 auto-sampler, Dionex Corp., Sunnyvale, CA), total organic carbon (dissolved carbon from organic sources that is available for microbial metabolic functions), pH, water temperature (°C) in 2009, conductivity (mS cm⁻¹), oxidation reduction potential (ORP, mVolts), dissolved oxygen, and mineral elements. The mineral elements (total P, K, Ca, Mg, Zn, Cu, Mn, Mo, Ni, Fe, S, Na, B, and Al) were analyzed via inductively coupled plasma emission spectrophotometer (ICP-ES, 61E Thermo Jarrell Ash, Franklin, MA). Quality assurance (QA) and quality control (QC) measures were insured per US EPA method 6010B (USEPA, 1997), method blanks and an ICP QC standard were checked every 10 samples. Detection limits for ICP were guaranteed onsite by the manufacturer for elements of interest to 5 ppb. All water sampling equipment was acid-washed, rinsed with MQ-water (ultrapure), prior to each

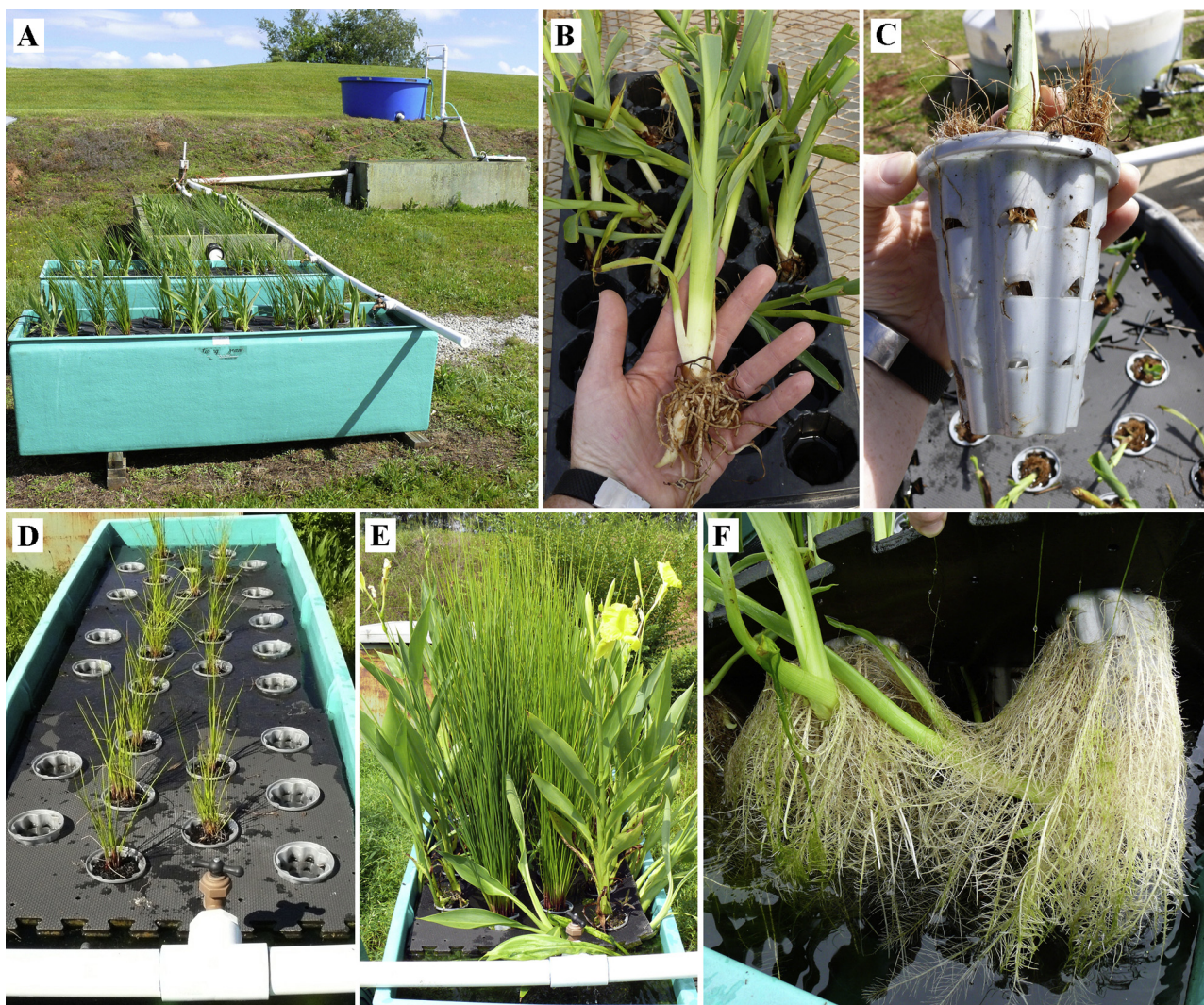


Fig. 1. The experimental floating treatment wetlands were established in replicated channel systems (A), using aquatic plant liners (B) seeded in aerator cups (C) that were placed in buoyant foam mats (D). After three months growth, plants were large (E) and plant root systems (F) were extensive.

sampling event. Final water samples were collected September 18 for the 2008 experiment and October 29 for the 2009 experiment.

2.4. Plant tissue samples

The roots (below-mat biomass) and shoots (above-mat biomass) of three plants per species per experimental FTW were harvested on September 18, 2008. Plant tissues were not harvested during the 2009 study. After harvest, the roots and shoots of each species were weighed (fresh mass), dried at 80 °C, weighed (dry mass), and ground in a Wiley mill (Swedesboro, NJ) to pass through a 40-mesh (0.425-mm) screen. Nitrogen concentration was determined using 100 mg of tissue and assayed using an Elementar Vario Macro Nitrogen combustion analyzer (Mt. Laurel, NJ) with tissue analysis procedures described by the Clemson University Agricultural Service Laboratory (2000). Phosphorus was assayed by the wet acid digestion procedure using the nitric acid and hydrogen peroxide method (CUASL, 2000). Phosphorus, K, Ca, Mg, Zn, Cu, Mn, Fe, S, Na, B, and Al concentrations in plant tissues were determined by ICP-ES with calibration standards rerun at midpoint and end of each analytical run.

2.5. Data analysis

All data presented represent the average value for each sampling event \pm the standard error of the mean. When appropriate, statistical analyses were performed using SAS v9.2 (SAS Institute, Cary, NC); analyses performed utilized PROC GLM. The influence of the treatment main effects on nutrient removal and nutrient uptake were determined using the *means* and *lsmeans* options of PROC GLM ($P < 0.05$).

3. Results

3.1. Aqueous nitrogen and phosphorus dynamics

The average daily N loading rate into the experimental FTWs varied between the two years the study was conducted. In 2008, the N loading rate averaged $145 \text{ mg m}^{-2} \text{ day}^{-1}$ (range: 22.2–331) while the N loading rate in 2009 averaged $320 \text{ mg m}^{-2} \text{ day}^{-1}$ (range: 142–606). Despite the approximate doubling of the N loading rate in 2009, and a regulator valve break (limited flow of nutrient solution from stock tank into mixing tank) during the 3rd week of May in 2009, the N removal efficacy of the experimental FTWs remained

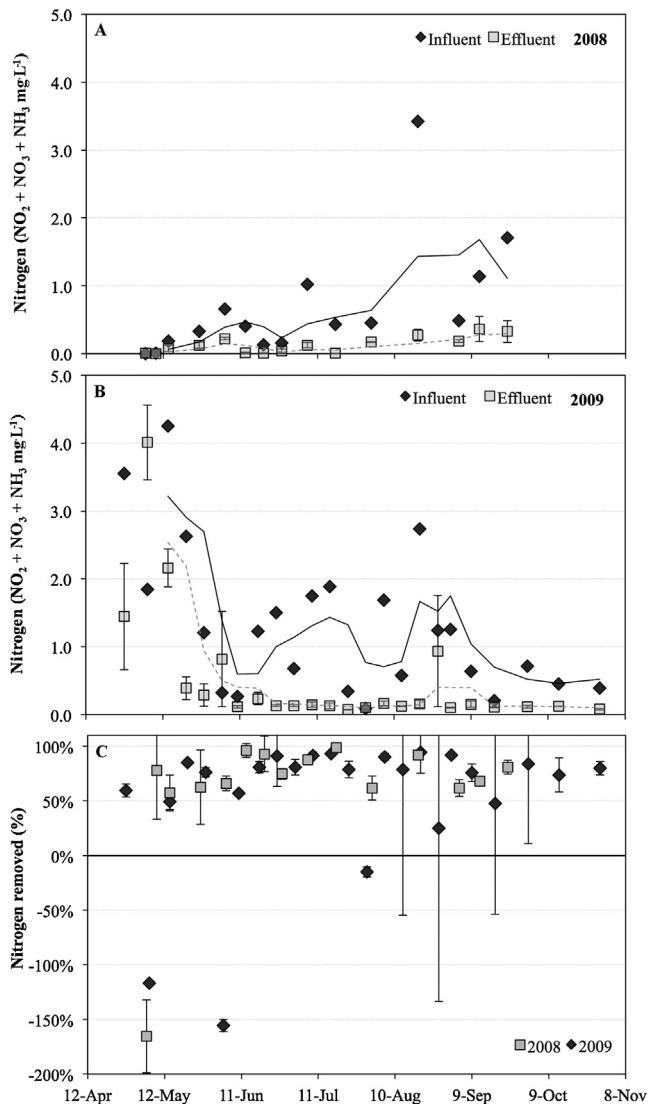


Fig. 2. Nitrogen concentration entering (influent) and leaving (effluent) floating treatment wetland systems in (A) 2008 and (B) 2009 and the resulting nitrogen removal efficacy (C). Sampling occurred on a weekly basis, data points represent sample means \pm the standard error of the mean, and are presented with 3-point moving average trendline.

positive (Fig. 2), and the percent daily load of N was reduced by 87.9% in 2008 and 66.9% in 2009 (Table 1).

Phosphorus loading rates into the experimental FTWs in 2008 averaged $13.4 \text{ mg m}^{-2} \text{ day}^{-1}$ (range: 3.5–26.4) and in 2009 averaged $37.2 \text{ mg m}^{-2} \text{ day}^{-1}$ (range: 20.4–50.1). The concentration of P flowing into the experimental FTWs in 2009 was $0.22 \text{ mg m}^{-2} \text{ L}^{-1}$ P, which was twice that of inflow concentrations in 2008. The average daily P concentration in effluent from the experimental FTWs was reduced by 75% in 2008 and 45.5% in 2009 (Table 1, Fig. 3).

3.2. Plant growth and nutrient uptake

Initial *Canna* shoot height was similar in 2008 and 2009. In the 2009 season, *Canna* plants were taller ($P < 0.0001$) and had longer root systems ($P = 0.0002$) at the time of harvest (Fig. 4). Initial *Juncus* shoot height was similar in 2008 and 2009. During the 2009 season, *Juncus* were significantly taller ($P = 0.0034$, Fig. 4); while root length was shorter ($P < 0.0001$). *Juncus* tolerated low nutrient

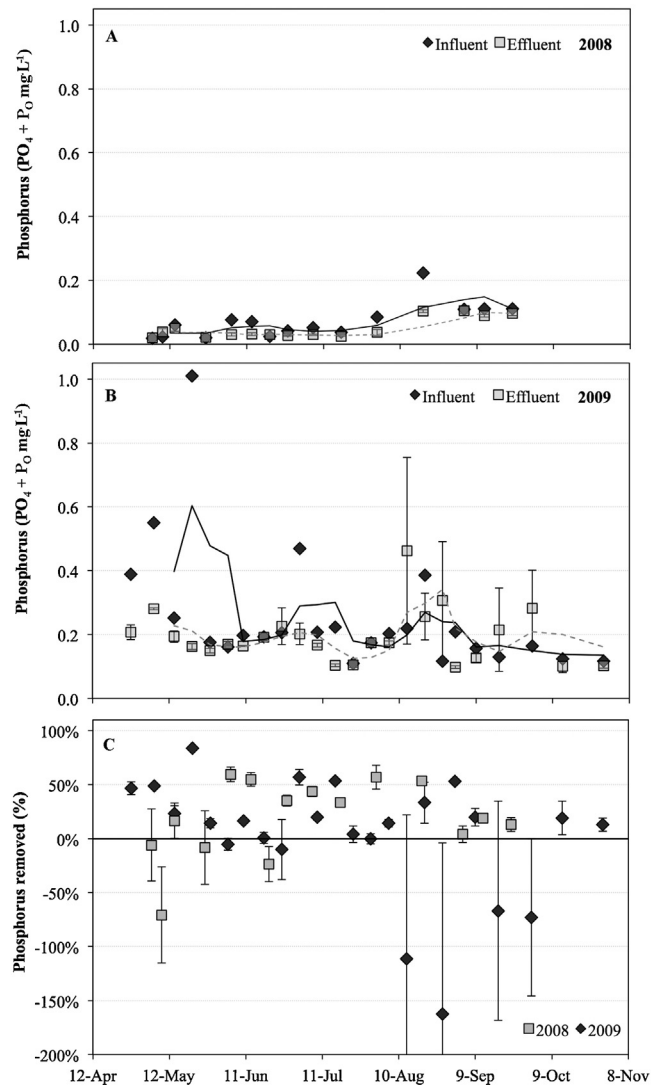


Fig. 3. Phosphorus concentration entering (influent) and leaving (effluent) experimental floating treatment wetland systems in (A) 2008 and (B) 2009 and the resulting phosphorus removal efficacy (C). Sampling occurred on a weekly basis, data points represent sample means \pm the standard error of the mean, and are presented with 3-point moving average trendlines.

loading rates better than *Canna*, as was evidenced by differences in growth and general plant appearance (data not reported) between plants in the 2008 and 2009 experiments. *Juncus* assimilated more N in both root ($P = 0.046$) and shoot ($P = 0.041$) tissues than *Canna*, when normalized on a surface area basis (g m^{-2} at harvest; Fig. 5). *Juncus* shoots also assimilated more P than *Canna* ($P = 0.034$) when normalized on surface area basis (g m^{-2} at harvest). The mass of P fixed by the root systems of both species was similar ($P = 0.1181$; Fig. 5).

3.3. Physico-chemical responses

Documenting changes in the physico-chemical characteristics of effluent leaving treatment systems is important as they can either positively influence ecological factors (e.g. temperature moderation), or negatively influence ecological factors (e.g. decreased dissolved oxygen or increased turbidity). During the two-year study, system pH, dissolved oxygen, and temperature (2009 only) were monitored. Influent pH decreased from 8.6 ± 0.11

Table 1Average daily loading^a of nitrogen and phosphorus into experimental floating treatment wetland units.

	Nitrogen		Phosphorus	
	2008	2009	2008	2009
<i>Daily load (mg m⁻² day⁻¹)</i>				
Influent	145 (54.6)	320 (64.7)	13.1 (4.16)	37.2 (7.90)
Effluent	17.5 (4.70)	106 (36.4)	2.5 (0.64)	15.4 (4.25)
Reduction in daily load	128	214	10.6	21.8
% Daily load reduction	87.9%	66.9%	80.9%	58.6%
<i>Average concentration (mg L⁻¹)</i>				
Influent	0.85 (0.32)	1.88 (0.38)	0.08 (0.02)	0.22 (0.05)
Effluent	0.14 (0.04)	0.79 (0.30)	0.02 (0.00)	0.12 (0.03)
Reduction in concentration	0.71	1.09	0.06	0.10
% Concentration reduction	83.5%	58.0%	75.0%	45.5%
<i>Mass balance (g m⁻² experiment⁻¹)</i>				
Total influent load	228	754	10.6	52.0
Total effluent load	13.9 (7.39)	144 (93.6)	1.69 (0.89)	17.8 (13.4)
Total load reduction	214	610	8.9	34.2
Plant uptake				
<i>Canna</i>	35.1 (6.00)	–	2.27 (0.44)	–
<i>Juncus</i>	60.6 (10.5)	–	3.71 (0.80)	–
Other removal processes	118	–	2.93	–

^a Mean values (standard error) of the mean.

to 6.2 ± 0.06 (effluent) before it was released from the treatment units ($P < 0.0001$, Fig. 6). Dissolved oxygen levels in the water column declined from an average of 7.7 ± 0.27 (influent) to 3.9 ± 1.5 mg L⁻¹ (effluent, $P < 0.0001$); approximately 50%. Similarly, in 2009, dissolved oxygen concentrations declined from an average of 8.5 ± 1.1 (influent) to 2.8 ± 1.0 mg L⁻¹ (effluent, $P < 0.0001$); approximately 66%. In 2009, the average water temperature recorded in the experimental FTW effluent was 23.0 ± 1.19 °C, was nearly 2 degrees lower ($P = 0.015$) than the influent temperature (24.8 ± 1.15 °C) entering the treatment troughs (Fig. 6).

4. Discussion

4.1. Aqueous nitrogen and phosphorus removal

The N removal percentages in this study are similar to those reported for surface and subsurface flow constructed wetland systems (Vymazal, 2007; White et al., 2010). A 2008 study by Yang et al. reported 71% (1 day HRT), 97% (2 day HRT), and 91% (3 day HRT) N removal from simulated agricultural runoff (3.8 to 7.9 mg L⁻¹ total N in influent) with 100% cover of FTWs planted with *Oenanthe*

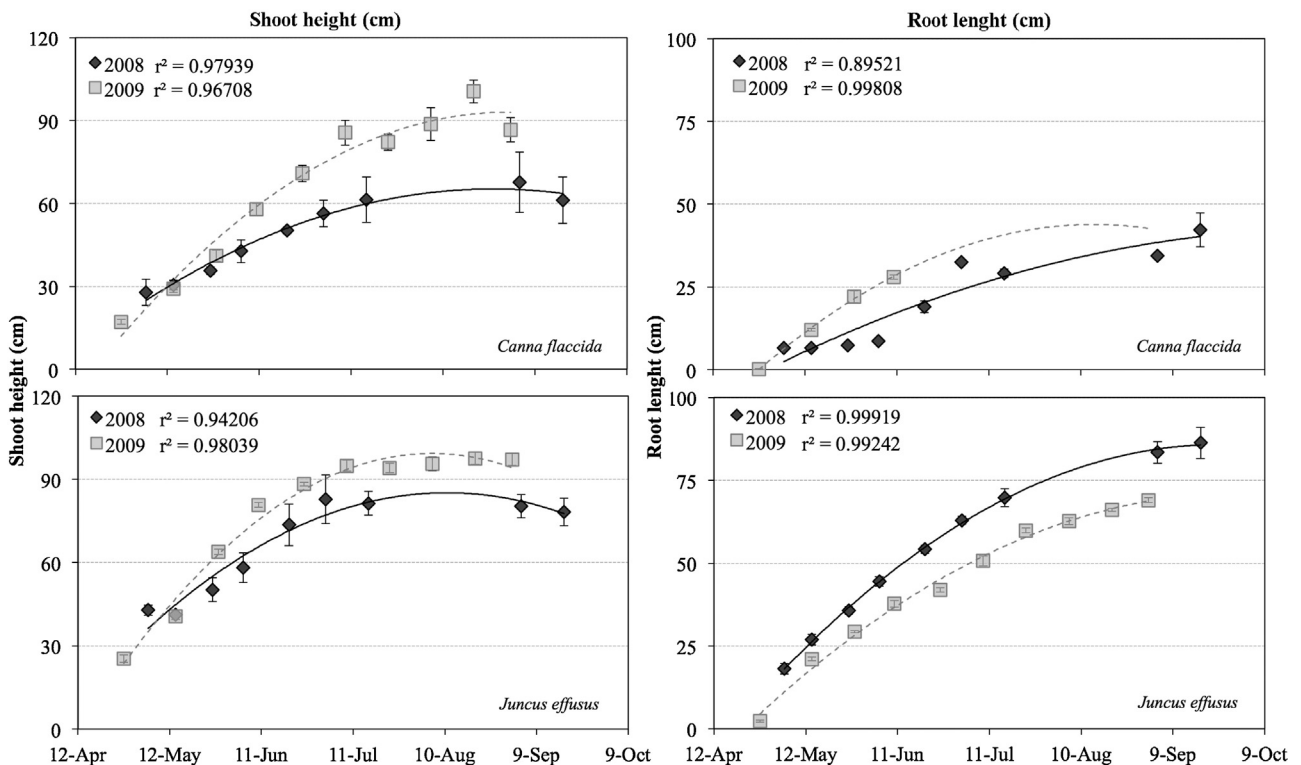


Fig. 4. Plant shoot (height) and root (length) of two aquatic plants (*C. flaccida* and *J. effusus*) established in the experimental floating treatment wetlands during the 2008 and 2009 growing season. Three replicate plants were measured per sampling period and experimental unit. Each point ($n = 18$) represents the average \pm standard error of the average.

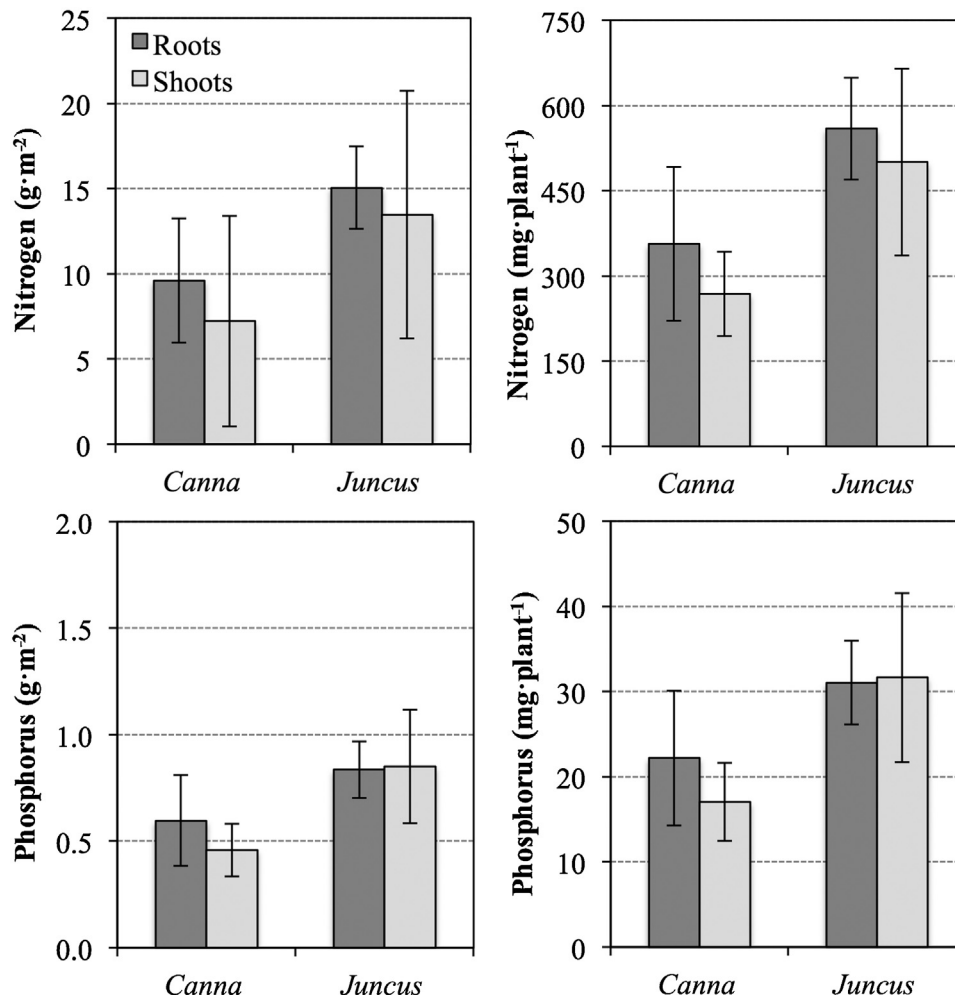


Fig. 5. Nitrogen and phosphorus above and below biomass of aquatic plants in experimental floating treatment wetlands. Nutrients are reported on both a surface area (m^2) and a per plant basis at the conclusion of the 2008 experimental season. Three plant replicates per bar, each bar represents the average \pm standard error of the average.

javanica (water celery). Stewart et al. (2008) reported FTW assisted nitrate removal rates of $985 \text{ mg m}^{-2} \text{ d}^{-1}$ when inflow concentrations were $230 \text{ mg L}^{-1} \text{ NO}_3\text{-N}$, a typical concentration for effluent from concentrated animal feedlot operations.

The range of P concentrations entering constructed wetland treatment systems described in the literature are from 0.7 to 10.5 mg/L P (Taylor et al., 2006; Vymazal, 2007), and processing by these systems has yielded effluent concentrations ranging from 0.02 to 5.15 mg/L P . Despite the low inflow P concentrations ($0.08\text{--}0.22 \text{ mg L}^{-1} \text{ P}$) in this study, removal efficacy averaged 45–75% (Fig. 3; Table 1). These results differed from those reported by Chang et al. (2012), Stewart et al. (2008), Tanner and Headley (2011), and Yang et al. (2008) who reported P removal percentages of 5–27%. Chang et al. (2012) utilized 5% or 10% cover of ponds with Biohaven™ FTWs. Biohaven™ FTW scaffolds are 17.5 cm thick fibrous (intertwined polymer strands) mats that are both porous and permeable, and have pre-cut holes, 12-cm in depth into which Canadian peat was added to provide a substrate for plant growth. Similar concentrations of P ($<1 \text{ mg L}^{-1}$) were used in the 2012 study by Chang et al., and reported P removal ranged from 5% to 20%. Yang et al. (2008) used higher concentrations ($1.25\text{--}1.54 \text{ mg L}^{-1} \text{ P}$) and P removal assisted by FTW averaged 13% (1 day HRT), 15% (2 day HRT), and 8% (3 day HRT); the authors' proposed that FTWs scaffolds needed an additional sorbent/substrate surrounding the root system of the plants in FTWs to adequately remove P.

Stewart et al. (2008) reported removal rates of $39.8 \text{ mg m}^{-2} \text{ d}^{-1} \text{ PO}_4\text{-P}$ in experimental FTWs with 100% cover with Biohaven™ floating islands. Stewart et al. (2008) hypothesized that aeration enhanced microbial activity within the rhizosphere of the FTW; thus, enhancing $\text{PO}_4\text{-P}$ removal by creating oxic zones within the rhizosphere. Tanner and Headley (2011) reported that planted Biohaven™ FTWs facilitated 3–27% dissolved reactive P removal when fine suspended sediment was simultaneously added to the simulated stormwater solution ($0.096\text{--}0.136 \text{ mg L}^{-1}$ total P). If fine suspended sediment was not present, then P was actually exported from the system.

Some of the differences in removal efficacy between the current study and the others discussed above may be due to FTW matrix effects (e.g. Beemats and Biohaven™), difference in percent of surface area coverage (range from 5% to 95% cover), differences in HRT, and/or study duration. The Biohaven™ FTWs evaluated in the four studies discussed above required substrate to support plants, while the Beemat system used in this study did not require substrate to support the plants. It is likely that the majority of P removal observed during this study occurred via plant uptake and entrapment in the microbial populations colonizing the plant rhizosphere, as suggested by Stewart et al. (2008). There were also substantial differences in the duration of FTW observation between this study and the four studies mentioned previously. The studies of Stewart et al. (2008) and Yang et al. (2008) were conducted over a

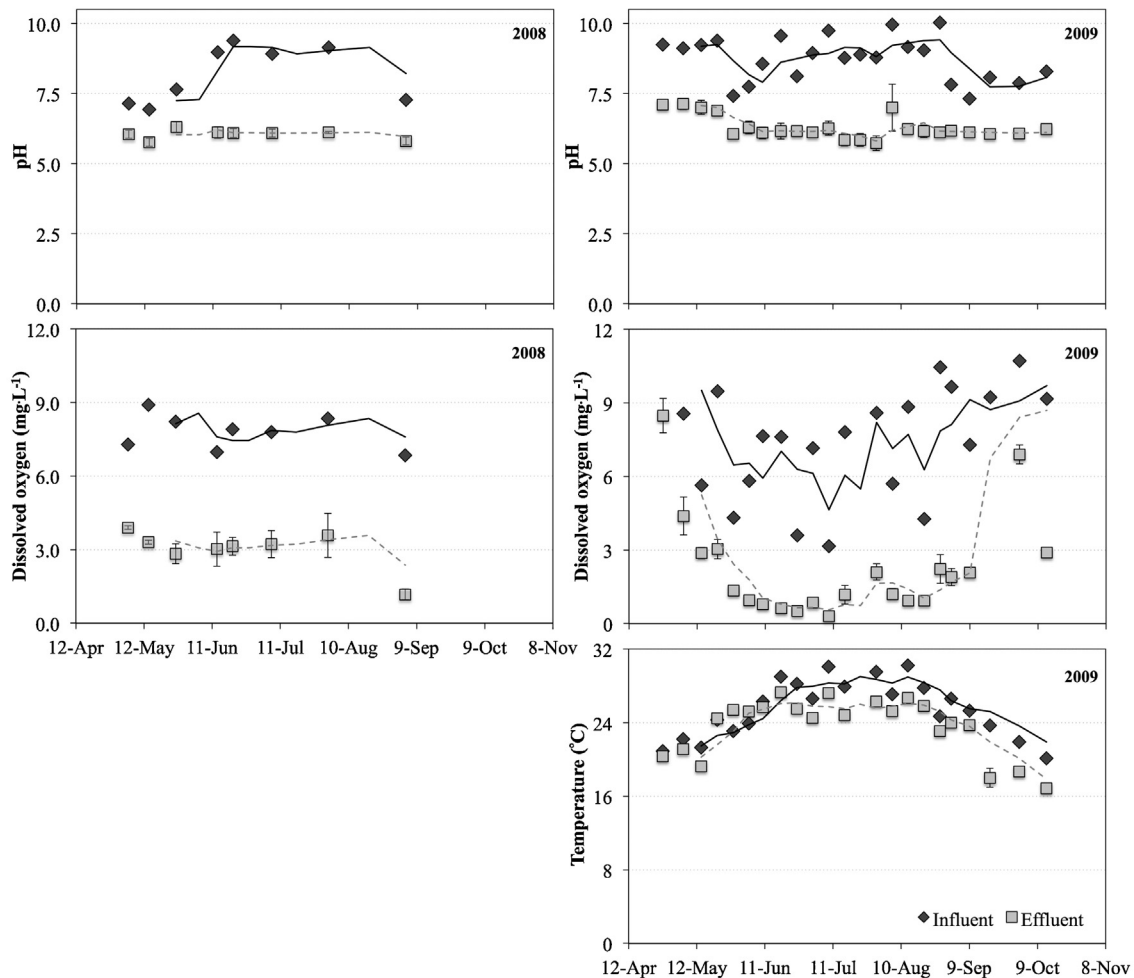


Fig. 6. Physico-chemical parameters monitored during 2008 and 2009. Samples were collected from influent control ($n=1$) and effluent of experimental floating treatment wetland units ($n=3$) on a weekly basis. Moving average trendlines for 3 consecutive data points are presented. Data points represent weekly sampling means \pm standard error of the mean.

15-day duration, the study of [Tanner and Headley \(2011\)](#) was conducted over 7-day intervals and this study evaluated FTWs for two, 5-month exposure durations (95% cover). This characterization of FTW impact on nutrient removal at two nutrient loading rates over two growing seasons provides added insights into consistency of nutrient removal with relatively short HRTs. Typical stormwater ponds may not turn over every 3 days, as were nutrient solutions in this study, thus nutrient remediation potential may increase with longer retention times.

4.2. Plant growth and nutrient uptake

Greater shoot growth of *Canna* and *Juncus* in 2009 was likely due to increased nutrient loading facilitating greater biomass accumulation. However, the root systems of *Juncus* were not as extensive in 2009, indicating a potential shift in sink strength (from roots to shoots). This shift in allocation strategy may have occurred because nutrients were plentiful; thus, available nutrient resources were allocated to foliar growth and biomass production rather than rhizosphere expansion ([Müller et al., 2000](#)). [Tanner and Headley \(2011\)](#) reported percent N and P accumulated in *Juncus edgariae* shoots (1.2% N and 0.17% P) and roots (0.9% N and 0.10% P). *J. edgariae* tissue accumulation was slightly higher than those of the *J. effusus* shoots (1.01% N and 0.06% P) and roots (0.82% N and 0.05% P; [Table A1](#)) in the current study. This disparity in accumulation could

be due to species differences or to differences in nutrient availability, as 7–8 mg L⁻¹ N were available to the *Juncus* in the [Tanner and Headley \(2011\)](#) study and only 0.85 mg L⁻¹ N were available for *Juncus* in the current study.

Juncus also accumulated more N and P in both root and shoot tissues than did *Canna* on both a per plant basis and on a per unit area basis. This difference in accumulation may be due to the relative inefficiency with which *Canna* accumulated nutrients into tissues in low nutrient environments. [Polomski et al. \(2007\)](#) reported that *Canna* efficiently accumulated N and PO₄-P from solution in high nutrient environments (>10.44 mg L⁻¹ N and 1.86 mg L⁻¹ P) and that in low nutrient environments (<1.75 mg L⁻¹ N and <0.18 mg L⁻¹ P), similar to the experimental exposures in our study, root tissues were greater sinks than shoot tissues for both N and P. *Juncus* may be more efficient at mining nutrients in the experimental low nutrient environment than *Canna*. This theory was supported by data generated through examination of estimated mass balance for nutrient loading and export from the experimental systems. *Juncus* accounted for 28.3% of N and 41.6% of P removal as aided by the FTWs, while *Canna* accounted for only 16.4% of N and 25.5% of P ([Table 1](#)), indicating that *Juncus* more efficiently accumulated nutrients during the 2008 experiment. Regardless of individual plant species performance, together both species accumulated 95.7 g m⁻² N from solution and 5.98 g m⁻² P over the 5-month growing-season in 2008, removing nutrient mass from

the aquatic ecosystem and reducing nutrient availability for primary producers. If FTWs are considered as a BMP, plant harvest is likely to be a critical component of FTW management.

The manner of harvest (shoot vs. whole plant) is also important. During this study and in the 2011 Tanner and Headley study, nearly half of the nutrients accumulated within the whole plant were accumulated within the plant root systems (below-mat biomass). Thus, harvest of shoots alone would not harness the full nutrient removal potential of FTWs, as senescing tissue that is not harvested would reintroduce nutrients into the water column. Whole-plant harvest should be considered an intrinsic component of FTW use for BMP applications, as considerable proportions of N and P accumulate in root systems; especially in low-nutrient environments when shifts in sink strength drive root growth and accumulation of nutrients within the plant root system.

4.3. Influence of FTW components on physico-chemical components

Alteration of effluent pH by vegetation installed in FTWs may be useful in treatment situations where elevated pH of stormwater runoff changes the water quality in the receiving water body. Part of the decline in pH may have been driven by degradation of organic matter by aerobic organisms, release of CO₂ from the root system during respiration, or more likely, exudation of organic acids from the plant root systems (Iamchaturapatr et al., 2007). Polomski (personal communication) reported that pH of subsurface-flow wetland experimental units established with *Canna* exhibited similar declines in pH from 7.2 to 3.9 during weekly experiments. Blossfeld et al. (2011) reported that the rhizosphere of *J. effusus* was acidified when oxygen was released from its root system.

The observed decline in dissolved oxygen likely resulted from a variety of factors. First, the plant root systems nearly filled the entire volume of the experimental FTW, increasing the surface area for microbial populations to colonize. Tanner and Headley (2011) reported a decline in oxygen saturation under FTWs and attributed the decline to respiration from living plant roots, dead plant roots, and microbial communities. The higher microbial respiration rates may have been fueled by organic acids exuded from the root system (Neori et al., 2000). Further, Ding et al. (2012) reported that acidification increased disruption of the crystalline structure of cellulosic biomass by breaking the lignin seal, increasing the quality of litter fueling denitrification. Acidification of the rhizosphere aided by the planted FTWs in the current study may have served to improve the quality of carbon available for microbial processing in a similar manner. The final factor contributing to the decline in dissolved oxygen was the nearly complete occlusion of the air/water interface by the experimental FTWs, which likely contributed to oxygen depletion by preventing the partial replacement of removed dissolved oxygen with atmospheric oxygen. Ahn and Mitsch (2002) suggested that reduced surface turbulence and increased shading resulted in decreased dissolved oxygen in mesocosm-wetlands. Though dissolved oxygen declined, even at the lowest recorded concentration (0.71 mg L⁻¹), conditions may not have been sufficiently anoxic to promote consistent denitrification. As in the Tanner and Headley (2011) study, it is likely that anoxic and anaerobic microzones persisted in the FTW root systems and their associated microbial colonies, permitting nutrient transformation during this study. Plant uptake was responsible for 45% of the N removed during this study. As a NO₃-heavy fertilizer source was used to spike the nutrient solution flowing into the FTWs, the primary pathway for removal of N was likely denitrification.

Declining dissolved oxygen concentrations in natural water bodies is of concern because a lack of dissolved oxygen can lead to

the creation of dead zones (Joyce, 2000). As a result, future studies should thoroughly evaluate dissolved oxygen dynamics in pilot-scale or large-scale FTW systems (<0.5 acres) to document that microbial organisms colonizing FTWs roots do not consume too much oxygen from the water column. The decline of dissolved oxygen reported in this study was likely magnified by experimental conditions (95% surface area coverage) and would not be observed in large-scale deployment of FTWs; both economic and ecological concerns would ultimately limit percent surface area cover of FTWs installed in large-scale treatment applications. Thus, the impact of FTWs on dissolved oxygen concentrations may differ when large-scale systems are studied. If excess consumption of dissolved oxygen is a real concern after scale-up, it is possible to pair FTWs with aeration devices to ensure that adequate dissolved oxygen is maintained in the water column to support aquatic micro- and macro-fauna. Preliminary research at Clemson University has indicated that aeration may also enhance plant uptake of nutrients (data not published).

Another potential use for FTWs is moderation of water temperature. Alterations in water temperature, whether above or below weekly or seasonal averages, can be considered a pollutant; just as nutrient or metal contaminants (Keller and Cavallaro, 2008). The experimental FTW systems in this study had 95% cover and a HRT of 3 days. Thus, the high percent cover and long retention time likely caused the temperature moderation observed. One 2002 study compared temperature moderation over two years in both mesocosm- and large-wetlands and reported a slight decrease in temperature of water from mesocosm inflow to outflow while the water temperature in a large wetland increased during treatment (Ahn and Mitsch, 2002). Ahn and Mitsch (2002) attributed the decrease in temperature in the mesocosms to water column shading. Though temperature moderation of 2 °C may not persist when FTWs are deployed as BMPs, the potential of FTWs to decrease water temperature should be considered if one of the contaminants impacting surface water health is temperature.

5. Conclusions

Floating treatment wetlands facilitated consistent N removal (58–83.5%) over the two growing seasons and relatively consistent P removal with 75% of influent P removed in 2008 under lower nutrient loading conditions and 45.5% of P removed in 2009 when nutrient loads doubled (Figs. 2 and 3; Table 1). Nutrient uptake by *Juncus* exceeded that of *Canna*. Additional work should be conducted to examine the remediation efficacy of FTWs with individual plant species and species mixtures. The uptake capacity of various plant species under a range of nutrient loading rates needs to be documented and the nutrient uptake models derived from this data verified, as harvest may be an integral part of managing FTWs to effectively remove nutrients. BMP parameterization should include management strategies along with potential for nutrient removal. Further work with FTWs may reinforce their usefulness in low nutrient environments, where desired outflows of total P are <0.05 mg L⁻¹, and where other remediation means would be cost prohibitive.

Acknowledgments

This research was funded by Clemson University and Beeman's Nursery who provided the floating mats and vegetation tested during the study. The authors wish to thank Brandon C. Seda and J. Brad Glenn for contributions in sampling and laboratory work. Technical Contribution No. 6160 of the Clemson University Experiment Station.

Table A1

Mean (standard error) plant tissue macro and micro-nutrient concentrations recorded in shoots and roots of *C. flaccida* and *J. effusus* at the end of the 2008 floating treatment wetland experimental trial.

		Nitrogen (%)	Phosphorus (%)	Potassium (%)	Calcium (%)	Magnesium (%)	Sodium (%)	Sulfur (%)	Copper ($\mu\text{g g}^{-1}$)	Iron ($\mu\text{g g}^{-1}$)	Manganese ($\mu\text{g g}^{-1}$)	Zinc ($\mu\text{g g}^{-1}$)
<i>Canna</i>	Shoot	0.87 (0.09)	0.06 (0.01)	2.07 (0.55)	0.51 (0.07)	0.54 (0.14)	0.45 (0.10)	0.12 (0.01)	9.33 (1.11)	433 (91.5)	373 (141)	53.2 (6.26)
	Root	0.92 (0.06)	0.06 (0.01)	2.58 (0.53)	0.50 (0.09)	0.44 (0.11)	0.52 (0.13)	0.11 (0.01)	8.33 (0.96)	492 (97.9)	781 (349)	42.4 (3.89)
	Whole plant	0.87 (0.07)	0.06 (0.00)	2.30 (0.42)	0.51 (0.07)	0.51 (0.11)	0.47 (0.10)	0.12 (0.01)	8.87 (0.70)	451 (88)	433 (63)	50.1 (4.21)
<i>Juncus</i>	Shoot	1.01 (0.06)	0.06 (0.01)	1.62 (0.46)	0.40 (0.07)	0.30 (0.12)	0.34 (0.11)	0.13 (0.01)	9.44 (0.50)	431 (64.4)	1385 (356)	41.9 (3.44)
	Root	0.82 (0.07)	0.05 (0.00)	1.47 (0.38)	0.35 (0.07)	0.45 (0.20)	0.35 (0.14)	0.14 (0.01)	7.89 (0.86)	304 (35.2)	219 (61.8)	48.1 (3.74)
	Whole plant	0.93 (0.07)	0.06 (0.00)	1.54 (0.32)	0.38 (0.06)	0.36 (0.13)	0.35 (0.11)	0.13 (0.01)	8.57 (0.40)	402 (44.1)	869 (228)	45.4 (3.12)

Appendix A.

See Table A1.

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